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Article

The Biodiversity Offsetting Dilemma: Between Economic Rationales and Ecological Dynamics

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Abstract: Although many countries have included biodiversity offsetting (BO) requirements in their environmental regulations over the past four decades, this mechanism has recently been the object of renewed political interest. Incorporated into the mitigation hierarchy in three steps aimed at avoiding, reducing and offsetting residual impacts on biodiversity arising from development projects, BO is promoted as the way to achieve the political goal of No Net Loss of biodiversity (NNL). The recent success of BO is mainly based on its ability to provide economic incentives for biodiversity conservation. However, the diversity of BO mechanisms (direct offsets, banking mechanism and offsetting funds) and the various institutional frameworks within which they are applied generate substantial confusion about their economic and ecological implications. In this article, we first analyze the rationale for the BO approach from the welfare and ecological economics. We show that both these frameworks support the use of BO to address environmental externalities, but that they differ in how they consider the substitutability issue and levels of sustainability with regard to natural and manufactured capital, and in how they address ecological concerns. We then examine the economic and ecological performance criteria of BO from conceptual and empirical perspectives. We highlight that the three BO mechanisms involve different economic and ecological logics and inherent benefits, but also potential risks in meeting biodiversity conservation targets. We lastly investigate the ecological constraints with respect to the BO practice, and economic and organizational limitations of the BO system that may impede achievement of NNL goals. We then reveal the existence of a tension between the economic and ecological rationales in conducting BO that requires making choices about the NNL policy objectives. Finally, this article questions the place of BO in conservation policies and discusses the trade-off between political will and ecological opportunities involved in the BO approach.

Keywords: biodiversity conservation; biodiversity offsets; ecological compensation; economic incentives; environmental policies; human well-being; natural capital; no net loss; substitutability; weak and strong sustainability

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1. Introduction

Over the last two decades, environmental policies have increasingly used economic incentives for biodiversity conservation as more efficient ways of achieving conservation outcomes than traditional approaches [1]. Seen as a way to provide economic incentives, the concept of biodiversity offsetting (henceforth BO) has recently enjoyed renewed political interest, and is endorsed in many political agendas [2]. Whilst BO requirements have been appearing in the environmental regulations of many countries since the 1970s (but rarely implemented in practice [3]), BO has recently re-emerged in biodiversity strategies across national and international policies as the main innovative tool for biodiversity conservation worldwide [4]. Embodying a regulatory requirement, BO is primarily incorporated by law into the mitigation hierarchy in three steps aimed at avoiding, reducing and offsetting residual impacts on biodiversity arising from development projects [5]. The purpose of BO is to provide ecological gains counterbalancing negative impacts on biodiversity. In a context of economic development, BO is considered the main way to achieve the goal of No Net Loss (henceforth NNL) of biodiversity, currently being a central political objective [6].

In practice, the BO principle encompasses three main mechanisms: (1) direct offsets, requiring developers to carry out compensatory measures themselves through restoration actions or acquisitions of natural areas in which appropriate conservation plans are implemented; (2) the banking mechanism, whereby a third party called a bank operator implements larger restoration projects ahead of future impacts, generating thereby offsetting credits for future needs of developers; and (3) offsetting funds, organized by certain environmental organizations (public agencies or non-governmental conservation organizations) in order to collect money from developers to carry out restoration actions or conservation projects [7]. How these different mechanisms are used and regulated depends on the legislation and the institutional environments of each country [8]. In addition to this, there are also voluntary offsets in which developers propose offsets outside legal requirements, but we do not propose to treat them in this paper.

The rationale for the BO approach is to achieve ecological outcomes in a more efficient way than in traditional political approaches, being commonly regarded as a market-based instrument (henceforth MBI) [9]. Yet, while BO is often regarded as an MBI for biodiversity both in academic and political spheres (especially in the form of the banking mechanism), some recent articles have shown that BO mechanisms do not really share the characteristics of market or market-like instruments, either in theory or in practice [10,11]. However, most academics and policy-makers still value BO on economic grounds, the revival of BO in environmental policies largely resting on its ability to promote economic incentives for biodiversity conservation. Overall, by allowing environmental outcomes to be achieved without limiting economic development, the concept of BO offers the promise of making economic development and growth compatible with biodiversity conservation [12].

In addition, the economic rationale behind the BO scheme has raised major concerns with both academics and conservationists especially with respect to ecological goals. For some, this approach gives rise to a commodification of biodiversity that tends to jeopardize biodiversity conservation instead of ensuring it [9,13]. These observations have raised questions about the limitations and risks of using the BO approach for biodiversity conservation. Recent reviews on BO emphasized the main conceptual and practical limitations involved in the implementation of offsets (e.g., specific problems associated with metrics, equivalence, timing, spatial, compliance, monitoring, *etc.*) [14,15].

However to date, the economic foundations of the BO concept have rarely been addressed in the scientific literature. Moreover, the diversity of the BO mechanisms and the various institutional frameworks within which they are applied generate substantial confusion about their economic and ecological implications. Far from being exclusively of academic interest, analyzing the economic rationales of the BO approach should yield a better understanding of the functioning of its mechanisms, and help in dealing with their economic and ecological limitations.

In this article, we conduct such an analysis in three parts. First we examine the overall economic foundations and rationales for the BO approach to biodiversity conservation from an economics perspective. Whilst it is possible to address the overall rationale behind the BO scheme, we assume that the performance criteria of BO will vary across the three different BO mechanisms. We address this issue in the second part in two complementary ways. First, from a conceptual perspective, we examine the performance criteria of BO according to NNL policy goals and the type of equivalence targeted. Then, from an empirical perspective, we conduct a systematic analysis of the three different BO mechanisms in three steps by first describing their functioning, secondly pointing out their economic and ecological benefits, and thirdly highlighting their main risks. Finally, in the third part of this article, we discuss the main economic and ecological structural limitations and challenges of the BO approach when it comes to meeting biodiversity conservation objectives.

2. Economic Foundations and Rationales for the BO Approach

The principal objective behind the BO approach is to maintain biodiversity so as to achieve NNL of biodiversity in contexts where biodiversity losses occurred from development projects. BO is thus primarily addressed through the legal framework of Environmental Impacts Assessment [16]. In order to obtain permits, and in compliance with the mitigation hierarchy, developers are required to take compensatory measures to offset their environmental impacts leading to provide equivalent biodiversity gains. In most countries, environmental regulations aimed at in-kind offsetting mainly targeting “like-for-like” equivalence. This means that offset projects are designed according to ecological outcomes through actions of restoration, rehabilitation, creation or preservation of species and ecosystems [7,17]. In this perspective, regulatory frameworks only take into account the ecological gains provided by BO projects, regardless of the social or economic impacts of biodiversity losses. Thus, the rationale for using the BO system is primarily ecological.

However from an economics perspective, biodiversity losses represent social costs that go beyond the purely ecological level and need to be taken into account. Concern about the costs of loss of biodiversity has increased in recent years, especially since the publication of the Millennium Ecosystem Assessment reports [18] and The Economics of Ecosystems and Biodiversity reports [19]. These studies provide global economic assessments of biodiversity and ecosystem services, offering a general framework to link biodiversity and human well-being. They help to recognize that biodiversity underpins human well-being. In such a way, environmental losses are regarded as negative externalities that represent major costs for society and tend to reduce human well-being [20]. In this perspective, the main problems highlighted are decision-makers' failure to take biodiversity losses into account in economic calculations, and the lack of policy tools encouraging the internalization of negative externalities resulting from biodiversity losses [19]. Following the finding that traditional approaches failed in meeting the expected conservation outcomes, economic incentives have increasingly been used by policy-makers over recent decades to address environmental concerns [2]. Incentive approach aims to encourage economic decision-makers to adopt good environmental practices by offering compensation or rewards to individuals in exchange for environmental services [12].

By combining a regulatory approach, through the polluter-pays principle, with an economic incentives structure, the BO approach gained increasingly credibility and interest in political spheres [21,22].

The BO approach, in addition to ensuring that the legal compensatory obligation is met, should provide three major economic incentives that will influence developers' behavior and encourage good environmental practices. First, because BO represents significant costs for developers, it should be an incentive for developers to limit their impacts on biodiversity. Based on the insight that rational actors will perfectly weigh the economic costs and benefits of making their choices, developers are expected to minimize offsets, thereby reducing the impacts on biodiversity from their development projects. Second, the economic rationale of developers should lead them to comply with their offsetting requirements in the most efficient way, by seeking effective conservation projects [17]. Thus, if BO implementation is well framed and controlled by regulatory bodies, developers should in turn implement the best environmental practices by choosing the most cost-effective way to meet their offsets requirements (e.g., by using biodiversity banks). Lastly, through the financial benefits provided by some BO mechanisms, the BO system may provide incentives for private or public stakeholders to invest in conservation actions for economic reasons. The BO system can therefore exploit additional sources of funding for conservation actions, and may open the way to large and expensive conservation projects that probably could not have been implemented otherwise [23].

From the legal perspective, BO basically allows the internalization of negative externalities by requiring developers to offset the environmental losses they are causing. However, the rationale for the offsetting approach and the kind of offsets required can be regarded in different ways depending on the economics frameworks considered, especially with regard to welfare economics and ecological economics [24]. The main difference between these frameworks lies in how they consider the degree of substitution between the different forms of capital, especially natural capital and manufactured capital [25], and the different kinds of sustainability they imply [26,27].

Welfare economics aims to find solutions to internalize negative externalities (resulting for instance from development projects) in order to maintain the level of social welfare. In the welfare economics framework, the hypothesis is one of weak sustainability, meaning that manufactured and natural capital can perfectly be substituted for each other: what matters for future generations is only the total aggregate stock of manufactured and natural capital, but not natural capital *per se* [26]. According to this viewpoint, there is high degree of substitution between manufactured and natural capital implying that natural capital may decrease as long as manufactured capital increases in accordance with maintaining human well-being. In this perspective, there is no reason to specifically preserve the natural capital [28,29]. However, what matters is to maintain the social welfare then expressed through the utility provided by the production of goods and services. Welfare theory is principally based on the Pareto-optimum principle which states "that social welfare is maximum when it is impossible to make anyone better off (*i.e.*, happier or in a preferred situation) without making someone else worse off (with initial endowments)" [30]. According to this principle, economic activity is limited because no development projects can be Pareto-optimal since they give rise to negative externalities and decrease the utility of at least one individual [31]. A solution to overcome this problem lies in the Kaldor-Hicks compensation principle specifying that "as long as the sum of the total benefits of the project is greater than the sum of the total costs, a share of the benefits can be devoted to offsetting the social costs to meet the Pareto-optimality condition" [32,33]. In such situation, the compensation principle states that a change (e.g., resulting from development projects) is socially desirable if the individuals who are gaining from the new situation provide offsets to those who are suffering the losses. Moreover, if the costs of the offsets are borne by the developers and included in the total cost of the development project, the offsetting mechanism enables the internalization of negative externalities. The level of the offsets needed is assessed on the basis of the expected losses of utility. If losses of utility are assessed in monetary terms, offsets could be provided by financial gains. In this case, losses of natural capital will therefore be offset by financial gains instead of ecological gains. Thus, even if the welfare framework supports the use of offsetting to maintain the social welfare, high degree of substitution between the different types of capital is assumed (including natural capital and manufactured capital) because losses of natural capital are replaced by gains in other forms of capital. In this case, this is not about "biodiversity" offsetting but rather "utility" offsetting. However, the weak sustainability hypothesis can be regarded as over-optimistic in the light of recent works especially the TEEB and MEA reports leading to a very paradoxical situation called "the paradox of the environmentalist" [34,35]. Whereas values of biodiversity have been shown and the need to preserve it to maintain human well-being, it might be possible for the society to fall into an irreversible and highly degraded state of biodiversity while human well-being continues to increase, at least in the short term. With regard to this situation, if social welfare is to be maintained, utility losses resulting from biodiversity losses must be offset by gains in biodiversity and not by financial gains. Therefore in this case, beyond the hypothesis of high substitutability between the different forms of capital, a strong sustainability approach should be considered even in the welfare economics framework [36].

Conversely to this framework, the ecological economics approach is basically based on the strong sustainability criterion. This approach assumes that natural capital is an essential production factor, thus considering that natural capital and manufactured capital are complementary and not perfectly substitutable. The strong sustainability perspective defines ecological sustainability as "the natural limits set by the carrying capacity of the natural environment (physically, chemically and biologically), so that human use does not irreversibly impair the integrity and proper functioning of its natural processes and components" [37]. According to this viewpoint, a decrease in natural capital cannot be compensated for by an increase in manufactured capital [38]. In this case, offsetting cannot be financial and must result in gains in natural capital to maintain its level. However, the strong sustainability approach raises the problem of choosing the critical natural capital to maintain, and the minimum threshold levels below which they must not fall [39].

In practice, the implementation of the BO principle in environmental regulations aimed at achieving the NNL objective takes the strong sustainability perspective. In the context of development projects, losses in natural capital must be offset by gains in natural capital, implying that natural capital and manufactured capital cannot be substituted for each other. Moreover, the setting of the NNL objective in environmental policies reveals public and political awareness of the need to preserve biodiversity. The NNL goal was set during George H.W. Bush's campaign in 1988 in the United States of America (USA), initially to limit wetlands destruction [40]. The NNL objective then spread throughout the world, more recently becoming a political principle endorsed by many countries [6]. This commitment reveals the recognition of the various values and merits of biodiversity (social, economic and ecological) and the importance of maintaining natural capital by preserving it. Moreover, in contexts of strong ecological uncertainties about current and future biodiversity states and changes, incomplete knowledge about optimal levels of biodiversity, and when ecological extinctions are difficult to forecast [41], setting the NNL objective then represents a precautionary approach. According to the NNL perspective, two social choices are possible: either it is decided not to destroy the biodiversity that needs to be maintained, or it is decided to continue the economic and social developments because of their utility despite their environmental damages, but in this case, compensations for the destruction of biodiversity are involved. In this perspective, the BO principle represents the only way to reach the NNL objective. However, the level and type of ecological equivalence required between gains and losses depend on the equivalence criterion set in the NNL policy. The strict like-for-like equivalence represents for instance the highest levels of sustainability under BO regulations [3].

However, depending on the goals targeted in the NNL policy, different types of equivalence and associated offsetting can be provided. Indeed, the types of equivalence targeted in the BO system are closely linked to the goals of the NNL policies [3]. In line with Quétier *et al.* (2014), we examine from a conceptual perspective the various types of equivalence linked to NNL goals, and the associated BO baseline and metrics used to assess losses and gains in biodiversity (Table 1). When the NNL goal aims to maintain the level of human well-being, equivalence is based on the utility provided by the different forms of capital. In this case, losses of natural capital can be offset by gains in another capital, *i.e.*, manufactured capital (line 1, Table 1). Monetary metrics can be used to assess losses and gains using cost-benefits analysis to assess benefits of development projects and costs of biodiversity losses. From this perspective, a weak sustainability approach is assumed as high degree of substitutability between capital (see section above).

Conversely, when losses of natural capital must be offset by gains in natural capital, BO requires an ecological equivalence based on a strong sustainability approach. However, we propose to better reflect the practices of BO policies by introducing a finer distinction between the different components of natural capital based on three main approaches to biodiversity that are: ecosystem services, functional, and individual with species and habitats [42] (lines 2 to 4, Table 1). We further argue that these different approaches to natural capital involve different types of ecological equivalence and BO approaches. NNL goal can relate to ecosystem services that need to be maintained to human well-being. In this case, offsetting aims to replace ecosystem services damages. Ecological indicators tied to the different categories of ecosystem services can be used to assess losses and gains in ecosystem services [43]. A functional approach to natural capital can also be considered, with BO aimed at maintaining NNL of ecological functions. Functional indicators can be used to assess losses and gains in ecological functions [44]. Lastly, through an NNL goal focused on an individual approach to biodiversity (*i.e.*, species or habitats), BO is aimed at maintaining populations or communities of species or specific habitats. Here, biological indicators can be used to assess species (vegetal or animal) losses and gains [15]. These different approaches to biodiversity involve different ways of looking at biodiversity in particular with regard to the recognition of the complexity of its dynamics and processes (see Section 4.1).

Table 1. Types of equivalence, offsetting and metrics across No Net Loss (NNL) policy goals.

	NNL Goals	Types of Equivalence	Types of Offsetting	Possible Metrics Used for Assessing Losses and Gains
1	Maintaining human well-being	Equivalence in utility	Forms of capital: losses in natural capital can be offset by gains in another capital (e.g., manufactured capital)	Benefits of development projects *versus* values of biodiversity losses assessed through cost-benefits analysis
2	Maintaining the level of ecosystem services that are beneficial to human well-being	Equivalence in ecosystem services	Offsetting aimed at maintaining the production of damaged ecosystem services by providing equivalent gains in ecosystem services	Ecological indicators of ecosystem services by category (regulation, support, provision, cultural) (e.g., presence of species providing specific ecosystem services)
3	Maintaining ecological functions	Functional equivalence	Losses of ecological functions are offset by gains in the same ecological functions (e.g., habitat for species)	Functional indicators (e.g., habitat area, density of vegetation)
4	Maintaining species and habitats	Individual-based equivalence	Offsetting aims to replace the same species populations or communities and habitats lost	Biological indicators (e.g., presence/absence, species diversity)

3. Economic and Ecological Analysis of the BO mechanisms Performance

NNL policies commonly target, in practice, functional and individual (species or habitats) approaches to biodiversity to conduct BO. As stated above, three different mechanisms can be used to implement BO: direct offsets, banking mechanism and offsetting funds. We argue that economic and ecological criteria performance will vary among these three mechanisms [45].

3.1. The Direct Offsets Approach

Direct offsets involve the implementation of BO by the party responsible for environmental damages arising from development projects. In the US legislation, this system is commonly called “the permittee-responsible mitigation”. Offsets are single conservation projects tied to a given development project. In this way, each offsetting measure is defined and sized case by case in relation to specific quantified impacts [14]. Moreover, offsets are primarily carried out near the impacted area [7].

From an ecological perspective, this proximity between the offsetting site and the impacted area, combined with defining each offset in terms of one impact, should help reach the NNL objective through better ecological and geographical matching between biodiversity losses and gains [17]. However, for this direct offsets system to work, offsetting measures need to be properly incorporated into local conservation projects. Studies have shown that single offsets implemented regardless of local conservation projects and without being incorporated into spatial and temporal planning can lead to conservation failures [45]. Single offsets can also result in small and isolated conservation projects leading to ineffective conservation outcomes [46].

From the economic and organizational perspectives, the direct offsets approach is considered as being inefficient. In the USA, at the end of the 1980s, two reports pointed to shortcomings in the application of BO through the direct offsets approach leading to biodiversity losses [47,48]. These failures actually revealed major organizational difficulties in the enforcement of offsets requirements through the direct approach. This inefficiency was mainly due to the high transaction costs generated by the implementation of biodiversity offsets in individual cases both for regulatory bodies and developers especially where large development projects are concerned [49]. Indeed, in the direct offsets system, the developers themselves are required to implement their offsetting measures. However, they generally do not have the necessary skills to conduct offsets that require significant expertise and specific knowledge. In this case, developers generally use service providers and experts to conduct their offsets, but this increases the financial costs of offsets measures. However, the level of expertise needed is highly dependent on the NNL goals and on the type of compensatory measure targeted [50]. For instance, preservation measures (*i.e.*, purchasing an existing natural area in order to preserve it) require lower levels of expertise and knowledge than restoration or creation actions [50]. However, the implementation of BO requirements through preservation actions raises strong concerns among scholars and conservationists in relation to the issue of the additionality of compensatory measures to meet the NNL goal [14]. In the USA, since BO regulations were reinforced, restoration measures have accounted for the highest proportion of direct offsets measures carried out (42% [51]); in contrast, such measures account for the smallest proportion of offsets in France (17% [52]). For the regulatory bodies responsible for the enforcement and monitoring of BO, offsets carried out through the direct approach generate high transaction costs too. This approach requires regulatory bodies to control and monitor as many offset projects as development projects, which generates significant transaction costs and makes it difficult to enforce offsetting liabilities [49,50]. In the USA, this inefficiency of the direct offsets system led to the implementation of the banking mechanism in early 1990s in response to these economic and ecological flaws [13].

3.2. The BO Banking Mechanism

The BO banking is an innovative organizational form that emerged to meet offsetting requirements. Through this approach, a third part called an operator carries out offsetting measures on behalf of developers by creating an offsetting bank. This bank is composed of biodiversity credits corresponding to ecological gains provided by the bank operator. These gains generally result from restoration actions conducted ahead of future development projects that have been checked and approved by regulatory bodies before being used to offset developers' impacts. Thus, an offsetting bank serves to offset several and various impacts arising from different development projects. One of the main differences between this approach and the direct offsets approach is the transfer of responsibility from the developers to the bank operator in conducting and monitoring the compensatory measures over time (note that the period of liability depends on national regulations; under US law it is forever because conservation easements are linked to offsetting banks, while in France, the average period is about thirty years [53]). Legal responsibility for offsets can also be transferred to the bank operator, but this too depends on the regulatory framework (transfer is possible under the US law whereas in France the developer retains legal liability [53]). The banking mechanism actually encompasses various schemes under different names according to regulatory and institutional frameworks and to the type of biodiversity targeted (e.g., species or habitat bank, wetland mitigation bank, biobank). The bank operator can be a private or a public organization or individuals, and the offsetting bank can be commercial or non-commercial [7].

From an ecological perspective, the banking mechanism is supposed providing better conservation outcomes than in the direct offsets approach. First, by pooling various small offset actions within a larger offsetting project, the banking mechanism better guarantees ecological and conservation successes [54,55]. Combining the offsetting liabilities of several developers over a larger area providing greater ecological benefits increases the chances of successful offsets coming from biodiversity banks. Moreover, planning in advance for offsets through the banking mechanism encourages the right choice of offset sites and actions to be made in relation to local conservation issues. In addition, the banking approach helps prevent temporary losses of biodiversity by providing biodiversity gains ahead of future ecological impacts [56]. Indeed, a major argument for the ecological benefits provided by biodiversity banking is the effective temporal and spatial strategy that the mechanism encourages, in terms of offset locations and types (*i.e.*, ecological actions) [45]. Advance checking and approval by federal agencies of the ecological results that the bank proposes to offset future impacts also limits offsets failures [57]. Lastly, depending on the level of asset specificity set by regulations, the banking system can even aim for significant environmental gains, leading to good ecological restoration projects.

However, although the banking mechanism was expected to ensure ecological success, many case studies have revealed difficulties and failures in achieving NNL of biodiversity [58,59]. These findings have led to an extensive academic debate on the relevance of such tools for biodiversity conservation [60,61]. On technical concerns, some studies revealed particular problems in relation to spatial issues and restoration results [62,63]. Contrary to the direct offsets approach, the banking mechanism necessarily implies off-site offsets and greater gaps between losses and gains in biodiversity, despite the definition of a specific service area (*i.e.*, a geographic area in which the bank can sell its credits; in the US wetland mitigation banking, this is usually a sub-basin area from 255 km^2 to 3544 km^2 in size, depending on the State [64]). Indeed, this mechanism requires a sufficiently large area for the offsetting market to function properly (increasing offsets demand), which tends to reduce the geographical equivalence between ecological losses and gains. Moreover, due to the possibility of offsetting for multiple impacts via the same environmental gain, asset specificity is decreased [50]. Thus, as biodiversity offsets are not sized and carried out according to one specific ecological impact, the ecological equivalence between biodiversity losses and gains is supposed to be weaker than in direct offsets. In fact, for the banking mechanism to function properly, biodiversity credits need to be sufficiently standardized to be equivalent to several ecological impacts.

From the economic and organizational perspectives, the BO banking mechanism is commonly regarded as an MBI both in academic and political spheres (although some academics have challenged its implementation as such [64,65]). In theory, MBIs are expected to reach any desired level of ecological objectives in the most efficient way (*i.e.*, at the lowest cost) if they are properly designed and implemented [66]. In the case of BO, the desired level corresponds to the NNL objective that requires impacts on biodiversity to be offset. Due to specific features of the banking mechanism, this approach is expected to be the most efficient way to implement offsets [67]. The banking mechanism can be also regarded as an economic incentive for both developers and bank operators (see Section 2). Developers and bank operators are expected to find the most efficient ways to carry out offsets [66]. The main advantage of the banking mechanism lies in the use of an intermediary, which greatly reduces transaction costs for developers and regulatory bodies [49]. Through this mechanism, developers transfer the costs of implementation, management and monitoring of offsets to bank operators strongly limiting their transaction costs. Even though regulatory bodies spend time setting up an offsetting bank, once the bank is operational, less time and work are required to check and monitor offsets than with direct offsets, due to economies of scale. Thus, regulatory bodies can ensure better control and monitoring of offsets, leading to better enforcement of environmental regulations [50]. As for bank operators, even though they bear the transaction costs tied to offsets, they can achieve cost-effective implementation of offsets by taking advantage of the economies of scale resulting from large offsetting projects [68].

Turning briefly to the social dimension, the banking mechanism fosters the development of partnerships between parties who do not usually interact. Implementing an offsetting bank leads various stakeholders (public, private, organizations or individuals) from different sectors (business, conservation, agriculture) to exchange views on environmental issues. The banking mechanism encourages them to communicate and share, in order to balance their different goals. However, the banking mechanism also raises social inequality issues. The main concern is that those benefitting from offsets are not those suffering as a result of ecological losses. The banking mechanism implies a spatial gap between the impact and offsetting areas, meaning that the people who benefit from compensatory measures are not likely to be those who suffer from the environmental damage [69]. Although this issue deserves to be explored further, it is beyond the scope of this article.

The performance of the offsetting market depends on two main parameters: (1) the enforcement of offsetting liabilities that defines the offset demand (type of credits and quantity); and (2) the rules of the biodiversity banking system that determine the supply of offset credits (mainly the definition of ecological and geographical equivalence criteria setting the limits of the service area and the degree of asset specificity required). These two parameters actually depend on the institutional and legal contexts behind the BO device. The performance of the banking mechanism is, in fact, highly dependent on institutional and political choices.

From an empirical perspective, following the reports highlighting the inefficiency of the BO system in the USA, in 2008 the government set up the Final Rule to "improve the quality and success of compensatory mitigation projects" [51]. The definition of the 2008 Final Rule greatly reinforced the legal liabilities connected with BO by defining a specific framework, improving the control and monitoring of offsets, and providing precise rules (e.g., definition of a standardized method to assess ecological equivalence, requirement for funds for long term management, setting up a conservation easement, better organization of the banking system, *etc.*) [64]. Through the 2008 Final Rule, the US government encouraged developers to use the banking mechanism to conduct their offsets due to the banks' ecological and economic efficiency. The strong decrease in transaction costs to developers allowed through the banking mechanism and the reinforcement of regulations gave developers a stronger incentive to use this mechanism to meet their offsetting liabilities.

3.3. Offsetting Funds

In the offsetting funds system, also called "in-lieu fee mitigation" under the US legislation, developers pay a fee to specific entities, which differ depending on regulations (government, public agencies, non-governmental organizations, municipalities or environmental organizations). This mechanism also involves a third party who collects money from the developers and takes financial and legal responsibility for the success of offsets [70]. In this mechanism, the link between financial transfers and ecological gains is less direct and clear. Studies of offsetting fund programs showed that these payments often result in poorly planned offsetting projects that do not provide sufficient ecological and geographical equivalences with impacts [71]. The performance criteria are less demanding than in the banking or direct offsets mechanisms (with regard to management funds, equivalence assessment methods, control and monitoring of offsets implementation) [72]. Moreover, offsetting fund programs often provide offsets well after impacts occur. Thus, in most cases, offsetting funds do not provide sufficient guarantees that NNL of biodiversity will be achieved.

Offsetting fund programs face a risk of underestimating the funds required to conduct and achieve offsetting projects, or even to failure to perform the expected ecological actions [73]. In addition, the unclear link between the impacts of projects and offsetting requirements may mean that developers fail to take biodiversity into account when planning development projects [7]. In practice, offsetting funds are rarely incorporated into regulatory frameworks; they are often accepted in exceptional cases or in addition to other offsetting measures [7].

4. Main Structural Limitations for the BO Approach in Meeting the Biodiversity Conservation Objectives

4.1. Ecological Limitations

- **Limitations in integrating ecological knowledge through BO practices**

A major limitation of the BO system is that it is primarily based on incomplete and imprecise scientific knowledge regarding biodiversity and conservation issues [17]. Most of the practice of BO implicitly rests on scientific knowledge in the fields of ecology, conservation biology and ecological restoration. These disciplines remain relatively young scientific areas and face strong uncertainties with regard both to understanding biodiversity and its dynamics, and to predicting how it will evolve in a changing world [74,75]. In this context, the ecological success of BO remains uncertain and hard to predict. Restoration actions in particular have yielded mixed results, revealing difficulties in the recovery of the whole targeted ecosystem, with substantial and unrecovered ecological losses [54,62].

In addition to imprecise scientific knowledge, the BO system is conceptually impaired by the inherent difficulty of applying the most recent advances in ecology and conservation biology. Indeed, due to technical and operational limitations (e.g., time and spatial constraints in conducting environmental impact assessments and designing appropriate offsets), BO is constrained in practice to taking partial account of ecological scientific knowledge. Actually, the BO process is hardly influenced by current progress in scientific ecology, and a major gap is likely to result from the continuous mismatch between recent ecological researches and how biodiversity is treated in the BO process [76,77]. For instance, BO tends to consider species and habitats as isolated and static features of the ecosystem. However, this approach ignores a decade of research in ecology that has demonstrated the importance of adopting an even more systemic approach to biodiversity (*i.e.*, accounting for biodiversity dynamics, ecological interactions and processes) to consider higher degrees of ecological complexity (we define here ecological complexity by the property of ecological systems to be structured by multiple links and ecological interactions, emergent processes and non-linear dynamics) (Figure 1). Biodiversity responses to disturbance, for example, often show non-linear and not instantaneous dynamics [78]. Besides, in adopting a temporally and spatially restricted view of biodiversity during the assessment of losses, BO ignores the biodiversity potentially present (so called “dark-diversity”) [79]. The BO process also hardly considers the dynamics of biodiversity that result from processes interacting at different spatial scales from local to global [80]. Moreover, extinction debts may also be expressed long after disturbances, especially for species with long generation times, posing a major challenge for biodiversity conservation and BO [81,82].

This complex view of biodiversity makes it clear that biodiversity cannot be reduced to some of its isolated components (e.g., species or habitat), functions or utility (e.g., by adopting the lens of ecosystem services).

Moreover, switching from a systemic approach of biodiversity in the BO process to more functional or services approaches involves strong reductions in considering ecological complexity and represents an incomplete and less accurate view of biodiversity (Figure 1). First, ecosystem services are based on ecological functions which are judged to be useful to humans [83]. Therefore, one ecosystem service can be provided by different ecological functions. A complex lattice of species’ interactions, functional traits and dynamics are involved in ecosystem productivity, with varying degrees of usefulness. Ecosystem services are therefore only partially related to certain ecological functions; but many ecological functions cannot be equated with ecosystem services and can even constitute disservices (e.g., pollination is a function equated with a service for fruit production but with a disservice if invasive species are pollinated). There is no need for biodiversity to provide various ecosystem services that can be actually accomplished by human technologies (e.g., construction of a water treatment plant to provide the service of wastewater purification, pollination by hand to provide the pollination service).

Second, adopting a functional approach gives an incomplete view of biodiversity, as various and different species and ecosystems processes are able to provide the same ecological function (e.g., habitat function, climate regulation function). Moreover, ecological functions heavily depend on the structure and properties of ecosystems and on their distribution in space and time, which influence ecological interactions and ecosystem resilience [84]. Major ecological studies have thus emphasized the importance of adopting an even broader approach to biodiversity dynamics, ecological interactions and processes, resulting in a more thorough description of biodiversity. However, the reduction in considering the ecological complexity through functional or services approaches does not reveal the difficulties in assessing ecosystem services or ecological functions of biodiversity. Indeed, this is not because taking ecological complexity account is less high than assessments of ecosystem services or functions are easier to conduct.

Overall, the BO system fails to integrate and account for the ecological complexity of biodiversity in practice. Consequently, the BO process actually has problems accommodating the extensive contributions from ecology, especially when assessing biodiversity losses and designing equivalent offsets. The risk is that these failures may lead to underestimating ecological impacts, with the resulting incomplete or poor definition of equivalent offsets. From that perspective, BO may result in net loss of biodiversity, being unlikely to achieve NNL of biodiversity [61].

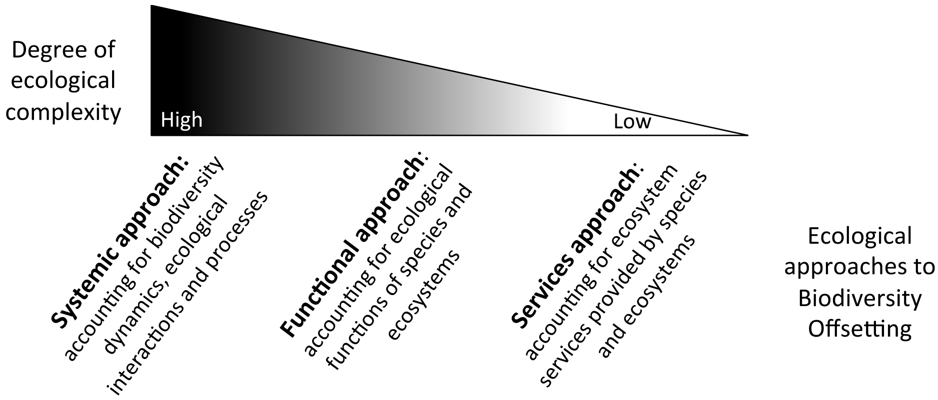

Figure 1. A complexity gradient across ecological approaches to biodiversity.

- **Limits to substitutability**

As Figure 1 illustrates, we argue that the degree of substitutability of biodiversity varies inversely with the degree of complexity of biodiversity (Figure 2). Mainly due to imprecise knowledge and technical difficulties both in accounting for biodiversity and in restoring ecosystems (as mentioned in the previous section), the higher the complexity of the biodiversity taken into account, the harder it is to reproduce the components of biodiversity, and thus to consider it as substitutable [85,86].

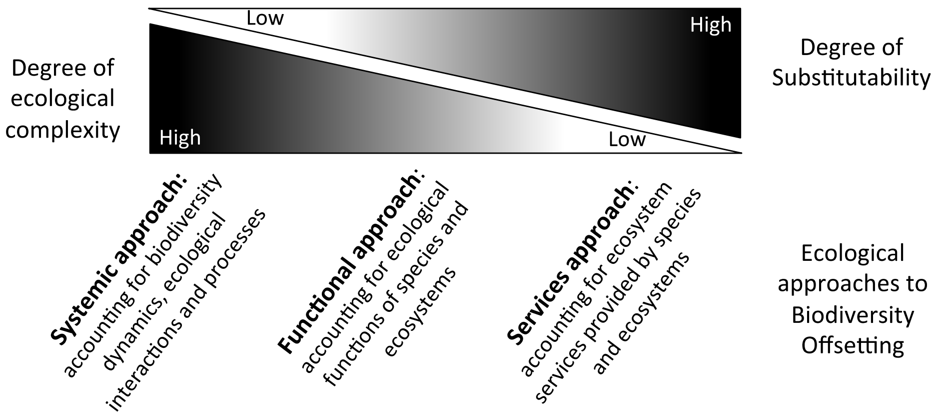

Figure 2. Trade-off between the degree of biodiversity complexity and its substitutability.

This leads to a paradoxical situation regarding BO. The NNL objective assumes a strong sustainability approach, which in turn implies low substitutability. However, the degree of substitutability depends on the ecological approach to biodiversity considered. It will be very difficult, perhaps impossible, to apply BO under a systemic approach to biodiversity because of impossibilities in accounting for all the biodiversity components. Thus, the NNL goal cannot be met easily when biodiversity is considered as being a highly complex object: the higher the degree of complexity involved in biodiversity losses, the less likely they are to be replaced and ecological equivalents found. And if we consider highly substitutable approaches of biodiversity *i.e.*, ecosystem services or functional approaches (Figure 2), we are failing to take the ecological complexity of biodiversity properly into account and this seems irrelevant from ecological perspective. This means that NNL policies and BO need to be defined and set up in such a way as to address the substitutability issues aiming at accounting the highest complexity gradient of biodiversity.

Placing the objectives of NNL policies within this context of substitutability constraints raises the question of NNL policy trade-offs. On the one hand, there is the will to implement an ambitious biodiversity conservation policy involving preserving a high degree of complexity in biodiversity, and on the other hand, BO risks falling short of the targeted results because of substitutability issues. In this situation, what kind of biodiversity should be targeted in NNL policies, and what are the most relevant ecological approaches to achieve it?

Answering these questions implies political choices based on the science available, and should, in our opinion, reflect a democratic choice of the kind of biodiversity we as a society agree to sacrifice and of the kind of biodiversity we decide to keep intact instead of pretending that win-win strategy can in most cases be found and scientifically supported. Indeed, the substitutability issue must be bounded by ethical concerns, based on social choice, for what we can offset and what we must preserve [87–89].

We suggest that the BO scheme be reserved for easily reproducible biodiversity *i.e.*, for ordinary biodiversity that includes many ecological equivalencies and allows considering simpler approaches of biodiversity due to lower conservation issues. The BO mechanisms cannot stand alone as a way to protect nature, but need to be backed up by properly-enforced public policy able to preserve non-substitutable ecosystems or species. Indeed, preventing the loss of biodiversity that we consider important to preserve from damage (such as endangered species and habitats) by strict statutory prohibitions remains the best way to guarantee no net loss of biodiversity.

4.2. Economic and Organizational Limitations

- **The risk of economic objectives prevailing over ecological objectives**

Through the use of economic incentives to preserve biodiversity, a new economic sector is emerging, featuring stakeholders new to the world of biodiversity conservation. This new economic sector involves, for example, environmental consultants, ecological engineering firms, companies and collaborative organizations whose primary aims are not necessarily those of biodiversity conservation. In the US where the offsetting sector is operating for a long time, the size of the overall annual market connected with BO is about USD 2.4–4.0 billion [8]. The BO system thus obviously encompasses more than purely environmental objectives; substantial lobbying now surrounds this market, with explicitly financial goals to be reached through "business solutions for a sustainable world" (http://www.wbcsd.org/home.aspx). The risk here is drifting away from a system aimed at preserving biodiversity towards a system aimed at ensuring economic outcomes [60]. This could have the perverse effect, as recent studies have shown [53,60], of encouraging some stakeholders to favor economic objectives over ecological objectives in order to preserve the link between the economic sector and the BO system.

- **The limited ability of economic design to meet ecological concerns**

While the banking mechanism is currently seen as the best way to perform BO from an ecological and an economic standpoint, this ecological and organizational efficiency carries risks. It could well encourage BO to be selected in preference to making initial efforts to avoid or minimize impacts on biodiversity. It needs to be remembered that the BO system necessarily implies ecological damage. Moreover, from an economic perspective, the proper functioning of this system implies assumptions on biodiversity assets. The banking system requires the most homogeneous and standardized biodiversity units to encourage trading of biodiversity credits. The more complex and specific the biodiversity credits, the harder it is for the offsetting bank to find buyers and sell its credits. In addition to the substitutability issues mentioned above, the economic mechanism behind the banking mechanism makes it difficult to target complex biodiversity with strong asset specificity. Even in the most effective mechanism, the banking system can thus lead to strong reduction in the complexity of biodiversity due to economic constraints imposing simplified biodiversity credits.

- **Organizational limitations: institutional risks**

In examining the BO mechanisms, we emphasized the importance of the institutional environment in ensuring good performance. The history of US legislation on BO shows that imprecise rules and the instability of the BO system leads in practice to offsetting failures. In fact, when an offsetting system is poorly designed and supervised, opportunistic behaviors can even lead to biodiversity losses. Nonetheless, some studies indicate that a certain flexibility needs to be maintained, for instance so as to allow the system to adapt to unexpected events resulting from environmental factors (e.g., species change in the context of climate changes) [90]. However, this institutionalization of the BO system depends on political will. Thus, under US and Australian legislation, the offsetting banking system is now well framed and regulated, leading to an improved BO system [49,50]. Better definition of the rules of the banking system encouraged bank operators to invest in conservation actions and developers to use the system. Elsewhere, especially in Europe, environmental regulations tied to BO liabilities have recently been significantly reinforced, but the banking system is still in experimental stages in most countries (France, United Kingdom, Germany) [8]. While this mechanism has begun to be introduced into environmental legislation, the design of current policy does not meet expectations because there are still many remaining institutional and organizational challenges to BO success.

5. Conclusions

The first objective of this paper was to clarify the economic background to the BO approach to biodiversity conservation. We showed that welfare economics and ecological economics offer relevant frameworks to analyze the BO approach. Whilst the basic assumptions behind these two approaches differ mainly in how they consider the substitutability and sustainability issues regarding capital, they both reveal the attractiveness of the BO concept and justify its use to address environmental externalities. However, they do not consider the same equivalence criteria and therefore do not involve the same performance criteria for BO. Welfare economics looks at equivalence in terms of utility, whereas ecological economics requires an ecological equivalence. Moreover, we showed that depending on the NNL policy goals, the BO system may consider different components of biodiversity involving different ecological approaches to biodiversity and resulting in different performance criteria and metrics used to assess losses and gains of biodiversity.

The second objective of this paper was to provide performance analysis of the three different BO mechanisms and highlight the main structural limitations of the BO approach in meeting biodiversity conservation objectives. Focused on ecological dimension, the banking mechanism ensures greater ecological effectiveness of offsets than the direct approach. However, in terms of ecological and geographical equivalence, the direct offsets approach is better at taking specific ecological features into account. From an economic perspective, the banking mechanism is more efficient than the direct offsets approach, but the economic constraints behind this mechanism can lead to inappropriate biodiversity conservation outcomes. Thus, defining a specific institutional framework and clarifying the regulations surrounding BO would appear to be crucial to the proper functioning of the system and the limitation of potential perverse risks.

Finally, we showed how the ecological limitations of BO point to a need to rethink NNL policies and BO goals in relation with the objectives of biodiversity conservation. In this sense, this paper offers a framework for debate on the balance between political will and ecological opportunities. Political choices are central to NNL policies, but they need to be based on the science currently available. In the light of ecological constraints, these choices will also necessarily involve public consultation.

Clearly, one of the main ways to improve the BO system is to better incorporate scientific contributions and social representation of biodiversity into the BO process. Ongoing ecological studies need to be used to support the increasing recourse to BO. Likewise, the scientific community should continue to investigate BO both in the natural and in the social sciences, providing both ecological and economics insights. While we have revealed the importance of economic factors within the BO process, careful investigation of how they operate in each specific project is needed.

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Author Contributions

Coralie Calvet, Claude Napoleone and Jean-Michel Salles conceived and designed the study; Coralie Calvet performed the research and wrote the paper. Claude Napoleone and Jean-Michel Salles also contributed to writing the paper. All three authors have read and approved the final manuscript.

Conflicts of Interest

The authors declare no conflict of interest.

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